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Lake and Stream Indicators for U.S. EPA's Environmental Monitoring and Assessment Program

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LAKE AND STREAM INDICATORS FOR U.S. EPA'S ENVIRONMENTAL MONITORING AND ASSESSMENT PROGRAM

January 24, 1991

By

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1.1
ABSTRACT

This paper summarizes the indicators proposed for a pilot study by the U.S. Environmental Protection Agency's (EPA) Environmental Monitoring and Assessment Program for lakes and streams (EMAP-LS). An additional lake and stream monitoring program is needed because current programs do not provide regional or national estimates of ecological condition or trends with known levels of confidence. Because national and regional monitoring must necessarily focus on spatial issues, we explain why and how we concentrate our sampling at particular index periods and in particular index locations. We provide 10 criteria for selecting response indicators and outline the 18 indicators selected and being developed for lakes and streams. Finally, we discuss how we propose to determine waterbody impairment through use of regional reference sites, regional experts, historical data, models, and empirical distributions of indicator values.

1.2
INTRODUCTION

Assessments of whether the conditions of the nation's waters are improving or degrading require data collected over long temporal and large geographic scales. Aggregating existing data from many independent monitoring networks has proven ineffective for this purpose because of the incomparability of their data over space and time. It has also been difficult to extrapolate findings from the sites sampled in such networks to national or regional populations of lakes and streams. This is because most existing monitoring programs in the United States lack an adequate statistical design for such objectives. Also, assessment of ambient biological conditions, in addition to the physical and chemical characteristics, of lakes and streams is needed. Where biological data are collected, they typically are filed and unused. These conclusions have been reached by several independent groups (GAO 1981, 1988, National Academy of Sciences 1977, Science Advisory Board 1980, U.S. EPA 1984, 1987b). To help correct the biological

shortcomings, the EPA (1990a) requires that states develop quantitative ambient biological criteria by 1996 to better determine ecological condition. But without some consistent methods of sampling and establishing criteria, results will not be interpretable from state to state, let alone nationally. The Environmental Monitoring and Assessment Program for lakes and streams (EMAP-LS) could assist this process by developing standard methods of sampling and data interpretation.

The legislative mandate for surface water protection is the Federal Water Pollution Control Act (P.L. 92-500) and its amendments. The objective of the act is to restore and maintain the physical, chemical, and biological integrity of the nation's waters [Sec. 101(a)]. Section 105(d)(3) requires EPA to develop and apply improved methods of measuring the effects of pollutants on the biological integrity of waters. Section 304(l)(A) requires listing of waters not expected to support balanced biological communities. Section 305(b) mandates biennial reports that assess the extent to which all waters provide for a balanced community of aquatic life. Trophic classification of lakes is required by Section 314(a). Finally, Section 319 requires listing of waters that cannot protect balanced aquatic communities without diffuse pollution controls.

Four current state and federal programs document aspects of surface water condition.

1. The state 305(b) reports, as summarized in the U.S. EPA report to Congress (U.S. EPA 1990b), estimated that 70% of the river miles and 74% of the lake acres assessed were reported to support designated aquatic life uses. These figures are based on assessments of approximately 29% of U.S. river miles and 41% of U.S. lake acres.

2. The National Contaminant Biomonitoring Program of the U.S. Fish and Wildlife Service is designed to document trends in contaminants that may threaten fish and wildlife populations. Fish from 112 river and Great Lakes stations are analyzed for 7 metals and 22 organic toxics. Schmitt and Brumbaugh (1990) and Schmitt et al. (1990) found significant declines in DDT, PCBs, selenium, lead, arsenic, and cadmium between 1976 and 1984.

3. The U.S. Geological Survey (USGS) monitors water quality at 412 National Stream Quality Accounting Network sites on major U.S. rivers and at 58 Hydrologic

Benchmark Network sites minimally impacted by humans. Smith et al. (1987) concluded that lead and fecal bacteria concentrations had decreased but nitrate, chloride, arsenic, and cadmium had increased from 1974 to 1981.

4. The USGS recently began a National Water Quality Assessment (NAWQA) program focused on describing and understanding water quality in a set of river basins across the country. Though similar to EMAP, NAWQA includes groundwater monitoring but not lakes, uses judgement-based site selection in a basin context rather than a national probability design, and focuses more on describing chemical, rather than ecological, conditions. Consequently, the NAWQA design may preclude estimating, with known confidence, the ecological condition of all surface waters in the nation.

Despite the millions of dollars spent by government agencies on monitoring and waste water treatment, biological condition remains a cause for concern. For example, annual Columbia River salmon and steelhead runs have declined by 75-85% or 7-14 million fish since 1910 (Ebel et al. 1989). The Tennessee River system has lost 20% of its mollusk species (Isom 1969), and 46% of the remaining species are endangered or seriously depleted throughout their ranges (Jenkinson 1981). Moyer (1986) reports that 38 states had fish consumption closures, restrictions, or advisories in 1985. Finally, nearly one-third of all North American fish taxa are endangered, threatened, or of special concern (Williams et al. 1989).

Although we collect a great deal of water quality data, we have little consistent understanding about national ecological condition or changes in condition. There is little knowledge of the reliability or uncertainty of available data. Apparently, we lack a quantitative framework from which we can produce reliable national or regional estimates of the condition of lake and stream ecosystems, especially in waters impacted by diffuse sources of pollution, biological introductions, and physical habitat alteration. This concern is reflected in EPA's Science Advisory Board report (Roberts 1990), which lists physical habitat alteration and biological diversity among the major ecological risks requiring attention by EPA. Brown and Roughgarden (1990) also reflected these concerns in their call for a U.S. Ecological Survey.

1.3
ENVIRONMENTAL MONITORING AND ASSESSMENT PROGRAM FOR LAKES AND STREAMS: MONITORING DESIGN

EMAP-LS, which is one of seven EMAP resource groups, is being developed to alleviate the above problems of making national and regional estimates of ecological condition from sites that may not be representative of the greater population of lakes and streams. EMAP-LS is designed to estimate, with known confidence, the ecological condition of regional and national lake and stream populations (Whittier and Paulsen In Press). This is accomplished by explicitly defining lake and stream populations and subpopulations, and by using an explicit method to list or identify them. A uniform grid adjusted by lake and stream density will be used to obtain randomized, systematic samples of surface waters with a geographical distribution reflecting that of the lake and stream populations. Nationally and for various ecological and political regions, EMAP is designed to determine:

- the number, proportion, and location of lakes and streams in various states of ecological condition,
- the proportions of those waters that are degrading or improving, and at what rates,
- the likely causes of impairment, and
- the general results of protective and restorative programs.

The EMAP design allows quantitative description of lake and stream condition within any regional boundary. If other federal agencies or states wish greater sampling intensity, EMAP can provide it, but currently the smallest sized regions for which we can provide estimates may approximate the size of the aggregate ecoregions of Omernik and Gallant (1990). They divided the conterminous United States into 16 major regions based on patterns of geology, landform, soil, vegetation, and general land use.

Monitoring will be organized into four tiers. Tier 1 currently consists of resource and landscape characterization from maps and remote sensing updated every 10 years for approximately 12,600 40-km^2 hexagons representing about 6% of the area of the conterminous United States. Tier 2 focuses on annual field sampling at approximately 800 lakes and 800 streams randomly selected from one-fourth (about 3,200) of the hexagons. A subset of these sites will be sampled each year to assess annual variation. Each year a different 1,600 sites will be sampled, returning to the original set every fifth year. Lakes and streams will be weighted by area classes to avoid overwhelming the sample with small water bodies. Tiers 3 and 4 involve sampling fewer sites more frequently, sampling a smaller area more intensively, or sampling ecological processes intensively at hand-picked sites. For example, we plan to seasonally monitor a set of 20-30 lakes and streams in the northeast U.S. to assess the results of reduced acidic emissions (Paulsen et al. In Press).

1.4
THE INDEX CONCEPT

To monitor 1,600 sites per year across the U.S., it is necessary to target times and places that are appropriate. We must also convert data on a number of indicators to information that is useful to nonscientists. To accomplish these goals, we use what we call an index concept. The index concept has three components: index period, index location, and numerical indices.

Index period refers to a sampling regime that targets one or a very few sample times, thereby limiting effort on individual units to maximize regional coverage with available resources. Measurements taken during an index period are intended as single snapshots of conditions to accompany snapshots from other waterbodies within the population of interest. Our interest is in the population of lakes or stream reaches, or the mosaic of snapshots, not the individual systems.

An ideal index period occurs when the values of the indicators are relatively stable. It must also be a time when indicator biota are present and measurable. If possible, the index period should be during a season of maximum environmental stress. In cases where the period of greatest stress is very short-lived or episodic, the index period should be chosen such that conditions may be related in a predictable way through models to the most stressful times.

The index period was applied successfully in the surface water surveys of the National Acidic Precipitation Assessment Program (NAPAP). The lake survey (Landers et al. 1987, Linthurst et al. 1986, Messer et al. 1986) employed a single sample taken during fall turnover. The relatively well-mixed fall turnover period was targeted to minimize spatial (particularly vertical) variability. The robust chemical classifications of lakes based on fall sampling can be related to the more biologically relevant acid stresses that occur during snowmelt (e.g., Eshleman 1988).

The NAPAP stream survey (Kaufmann et al. 1988, in press) indexed stream chemical conditions by sampling during spring baseflow after snowmelt, but before leafout. This serves as an index period for streams in acid-sensitive areas because it is a prolonged time of relatively low pH, coinciding with the sensitive life stages of fish species of concern. Chemical classifications of streams based on spring baseflow can be related to conditions of extremely low pH that occur during spring storm events, but cannot be monitored in a synoptic survey (Eshleman 1988).

Our index period will be mid- to late summer for lakes in the northeast U.S. Other times may be optimal for addressing certain topics but present their own problems. Accessibility and sampling crew safety eliminate the late fall to mid-spring period in northern states. Late fall and spring would miss times of greatest stratification and oxygen stress and lowest flows. The mid- to late summer period will certainly miss peak insect emergence, fish spawning, and spring and fall algal blooms. However, it appears to be an adequate index time to characterize regional populations of sites because it is

a period of considerable natural stress and the biological assemblages are more stable then than in other seasons.

An index location should provide in a single measurement (or an integration of relatively few measurements) adequate information about each lake basin or stream reach for classifications, regional extrapolations, or trend analyses. Multiple sampling points may be used to incorporate spatial variability of the indicator measurements. Alternately, measurements may be made at locations that themselves integrate the ecological stresses impinging on a waterbody as a whole. In a program that uses multiple indicators, it is not possible to choose an index location that is optimal for all indicators, and all possible stresses. However, it is necessary that the sampling times, locations, and protocols be adequate to express ecological conditions.

In EPA's National Lake Survey (Landers et al. 1987, Linthurst et al. 1986), samples were taken 1.5 m below the water surface at mid-lake to index nonlittoral lake water. In the National Stream Survey (Kaufmann et al. 1988, In Press), samples were taken mid-channel at opposite ends of a stream reach that was defined by confluent streams.

The large variety of lake and stream sizes, types, substrates, and morphologies offers multiple challenges for selecting appropriate index locations within lakes and streams, sampling techniques, and assessment methods. Some of this complexity may be stratified by choosing subpopulations for sampling and analyses. Our current thinking is to use the deepest area in a lake as the index location for water column and sedimentary indicators and to focus on major (>75% of the shoreline) littoral and riparian habitats for the other indicators. Stream sampling will also focus on major habitat types in a reach.

We will use indices to aggregate, synthesize, and convert data into information that is interpretable by decision makers and the public. Multimetric indices, such as the index of biotic integrity (IBI, Karr 1981), and habitat or trophic guilds are less variable than individual measurements (Landres 1983, Plafkin et al. 1989). Guilds or indices also

provide insight into the trophic structure of assemblages (Paine 1980) or other assemblage level properties that are less evident at the population (Grossman et al. 1990) or indicator species (Landres et al. 1988) level. At the same time, the component metrics or guilds contributing to an index allow us to examine what aspects of an assemblage or suite of stressors explain most of the observed change and how this changes from area to area.

In evaluating assemblage persistence or change it is also critical to consider the level of numerical resolution (Rahel 1990). Analyses of species presence/absence, proportionate or relative abundance, or absolute abundance data produce markedly different conclusions, as the papers stimulated by Grossman et al. (1982) indicate. EMAP-LS will focus on proportionate abundances because presence/absence data and relative abundances (rare, common, abundant) are too insensitive to change and absolute abundances are too variable.

1.5
LAKE AND STREAM INDICATORS

1.5.1. Conceptual Framework for Indicators of Condition

Assessments of surface water condition must relate to values and problems of present or potential concern to the public, regulators, and scientists. These issues are referred to as endpoints of concern (Hunsaker and Carpenter 1990, Hunsaker et al. 1990, Kelly and Harwell 1990, Suter 1990), and they reflect the ways lakes and streams are managed. For EMAP-LS, the endpoints are trophic condition, fishability, and biotic integrity. Other endpoints (esthetics, water supply, navigation, swimming) were considered but rejected because they are much less dependent on ecological condition, do not relate to prevailing ecological concerns, and are poorly associated with the ecological integrity objective of the Federal Water Pollution Control Act. By trophic state we simply mean the various classes from oligotrophic to dystrophic, plus general water clarity and appearance.

Fishability refers to the relative abundance, size, sustainability, and edibility of game fish. Biotic integrity is defined as a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region (Karr and Dudley 1981). Expectations for trophic state and fishability, as well as the measurements and indicators for all three, may vary regionally. Our assessments of condition will be made in terms of each of these endpoints, rather than the single entity of surface water condition (Paulsen et al. In Press).

Ultimately, we are concerned with the suitability of lakes and streams for the development, organization, reproduction, persistence, distribution, and evolution of successive generations of organisms, populations, species, and communities in response to different conditions (Warren et al. 1987). However, neither ecological health (as a single entity analogous to human health) nor the status of the endpoints of concern can be assessed directly (Karr 1981, Landres this volume, Schaeffer et al. 1988). Health assessments are initiated by making a series of measurements. Yet, by themselves, the raw data from these measurements are not sufficient. These data must be evaluated within a framework of ecosystem theory and known hazards to surface waters, and integrated or synthesized into information useful for decision makers, what we call indicators of condition.

EMAP has four types of indicators of ecological condition: response, exposure, habitat, and stressor. It is important to understand that these categories provide a conceptual framework, acting as a tool to guide the process of selecting, evaluating, and implementing the actual measurements used to assess ecological condition. These categories are not exclusive; a measurement or indicator may serve as more than one type.

1.5.2. Response Indicators

Response indicators are derived from measurements that describe the biotic condition of organisms, populations, communities, ecosystems, or other biological components or

processes of aquatic ecosystems as they relate to the endpoints of concern. Response indicators are the focal points upon which the health of lakes and streams will be estimated. The measurements should quantify the integrated response of ecosystem components to individual or multiple stressors. Response indicators should clearly relate to valued aspects of the environment, such as productivity, biodiversity, or sustainability. Response indicators may also be chosen to identify emerging problems.

EMAP-LS has several general criteria for evaluating the usefulness of response indicators (Herricks and Schaeffer 1985, Hughes and Paulsen 1990, Kelly and Harwell 1990). The indicator should be biological, incorporating elements of ecosystem structure, function, and process as well as aspects of population dynamics. The selected indicators should correlate with changes in other unmonitored biological components. They must have clear connections with the three endpoints of concern, and must be responsive to a broad array of potential stressors. An ideal indicator is applicable in a broad range of surface water types across the nation and serves as an early warning of deleterious change. The response indicators must be sensitive to varying levels of stress, but not produce excessive false alarms. This includes sensitivity to important episodes that occur outside the sampling period. Useful indicators are cost effective, supplying considerable information in a limited amount of sampling time. They should be easily implemented and yield reproducible results with low sampling variability. Initially, at least, their usefulness should be demonstrated from a previous monitoring program.

Schindler (1987, 1990) and Ford (1989) found that structural indicators, such as species, responded faster than process indicators, such as primary production, to acidification and eutrophication. This is largely because of the great amount of process redundancy among species. Of course, if sensitive species are community dominants or keystone species, process indicators would change with equal rapidity. Ultimately, the issue revolves around whether we are concerned with protecting processes or species. For the above reasons, we propose to focus on structural indicators but continue to evaluate possible process indicators. A simplified example of condition assessment using

structural response indicators is illustrated in Figure 1. The rectangles represent states of information. The diamonds show processes of refining information. Raw data from a collection of measurements (e.g., fish species and abundances, and individual lengths, weights, and general appearance) must be evaluated within an ecological context (waterbody type and size, season, geographic location, etc.) that defines what is expected (metric scoring criteria) for similar waterbodies. These evaluations produce metric scores (e.g., native species richness, % anomalies, % sensitive species). The metric scores may be further aggregated into indices like the IBI, or the index of well being (IWB, Gammon 1976). This process may range from simply summing unweighted metric scores to including metric scores in complex models. Both metric scores and index scores are used as response indicators in the process of deriving an overall statement of condition. The complexity of information (amount of detail) is reduced at each step of the process. Using both metric scores and index scores, rather than either alone, preserves more of the detail held in the raw data, which will also be analyzed and retained. However, use of metric scores may increase indicator variability and the possibility of contradictory assessment of condition, because metrics are not always highly correlated.

Our current 18 indicators are outlined in Table 1 with examples of their use in Tables 2 and 3. The metrics and scoring criteria in Table 2, though provisional, are presented to indicate our interest in developing metrics and evaluating particular metric scores.

1.5.2.1. Fish assemblages will be used to address two of the endpoints of concern, biotic integrity and fishability. The overall assemblage structure (proportionate abundance of each species, and individual size and general appearance) will be used to evaluate biotic integrity, while the sports fish portion of the assemblage will be used to address fishability. In some regions and stream types, fish assemblages (as measured by species proportionate abundances) are remarkably stable interannually (Matthews 1986, Ross et al. 1985) and over the summer index period. Matthews (1990) concluded that spatial

variation exceeded summer temporal variation in dominant stream fishes, but Grossman et al. (1990) found high coefficients of variation when examining absolute abundances of all species of stream fishes interannually.

Well-established sampling methods exist (Hocutt and Stauffer 1980, Nielsen and Johnson 1983) but they are labor intensive, especially in lakes. Generally, lake sampling will be done with overnight sets of gill nets in deeper water, traps near shore, beach seining, and night electrofishing in the littoral zone. Sampling will focus on major habitat types (deep and shallow water, cover, beaches, shoals). In most regions, streams will be sampled by electrofishing or seining but additional gear will be investigated in rivers. Voucher specimens will be archived in regional ichthyological museums.

A number of options for data interpretation exist, including species richness (Whittier and Rankin this volume) and various metrics and indices, such as IBI and IWB (Hughes and Gammon 1987, Ohio EPA 1988). As a data interpretation technique, the IBI has proven useful for stream fish assemblages in the Midwest (Karr et al. 1986, Steedman 1988), and has been modified for use in several other regions (Miller et al. 1988, Oberdorff and Hughes In Press). To examine how it might be used to assess fish assemblages in Midwestern streams, we analyzed Ohio EPA's data base, which consisted of several thousand collections over the past 10 years. Sample standard deviations (s) in IBI vary from 10 to 13 IBI units for the entire state, but include known regional differences and an improving trend in fish assemblages. Subtracting these, or among-site variation, leaves an s in IBI scores of approximately 5-6 IBI units. This standard deviation places us between the two lower curves in Figure 2, which is based on the nomogram of Loftis et al. (1989, Figure 2). Thus, with 50 subpopulation or regional sites we should be able to detect a change of 4-6 IBI units, rather than the 7-9 units discernable with an s of 10-13. Twenty sites at the lower s would allow us to detect a change of 6-9 units. Given a national sampling of 800 stream sites per year and 50 sites per region we should be able to discern changes in IBI scores of less than 10% in 16 regions.

Currently, there are no indices of biotic integrity for fish assemblages in lakes or in most fish depauperate regions or for fishability. Such indices are research tasks for EMAP-LS. Until these indices are developed, biotic integrity will be assessed by metrics such as species richness, % sensitive species, % exotic species, size structure, and rates of external anomalies.

Fishing quality is a very important endpoint of concern for lakes and streams. Fishability will be assessed by metrics such as game fish species present, relative abundance of game fish, age/size structure of game fish populations (Are there any large game fish? Are populations reproducing?), presence of external anomalies and disease (Do the fish look edible?), and state or federal consumption criteria violations (Are the fish actually safe to eat?). A fishery questionnaire similar to that in Plafkin et al. (1989) may also be used, but direct creel censusing is not cost-effective given the short time available per site.

1.5.2.2. Benthic macroinvertebrate assemblage structure will also be used to evaluate biotic integrity. Macroinvertebrates play a significant role in stream carbon processing and in the food web between the primary producers and fish, and are thus important to biotic integrity (Resh and Rosenberg 1984). Macroinvertebrates are a particularly important class of organisms to monitor in those systems that naturally have very few or no fish. In addition, this group has been used extensively as a biomonitoring tool to assess the effects of point and diffuse sources of pollution in streams (Fiske 1988, Lenat 1988, Metcalfe 1989, Plafkin et al. 1989).

There is probably no best period to sample macroinvertebrates because the various taxonomic groups mature and emerge at different times throughout the year. Thus, any index period will miss some species present at other times. As with fish, the complexity of possible substrate types and lake and stream types makes selecting appropriate index locations and sampling methods difficult. To accommodate lake and stream heterogeneity, we plan to sample a representative variety of each major habitat type

(profundal sediments, macrophytes, rocks/snags in lakes; pools, vegetation, rocks/snags, riffles in streams).

Lake sampling will probably be done with a combination of several methods, such as sweep nets, handpicking from natural substrates, and Ekman or Ponar dredges in littoral areas. Corers or dredges will be used in deep waters. Collection of sediment cores from the deepest area of the lake and determination of invertebrate remains will provide a complete record of present and previous chironomid assemblages, as well as other invertebrate groups (Walker 1987). Our sampling focus in streams will initially be on stable substrates (riffles, snags, vegetation). Sampling will probably be done with a combination of sweep and kick nets, and handpicking from natural substrates. Voucher specimens will be archived in regional museums.

Currently, there are a number of indices of biotic integrity for macroinvertebrate assemblages in streams. However, considerable research is needed to further evaluate these indices and to determine the sensitivities of species to various stressors in many regions. Candidate indices and metrics include: number and proportion of Ephemeroptera/Plecoptera/Trichoptera (EPT) taxa, taxa richness, proportions of certain taxonomic groups (e.g., EPT, Chironomidae) and feeding guilds (scrapers, shredders), similarity indices, and the Biotic Index (Cummins and Klug 1979, Lenat 1988, Hilsenhof 1987, Plafkin et al. 1989). We are unaware of similar indices of biotic integrity for macroinvertebrate assemblages in lakes, but we propose to evaluate and modify the stream indices for use in lakes and incorporate information about the tolerances of various lake taxa.

1.5.2.3. Zooplankton assemblages will be evaluated to assess biotic integrity and fish food availability. They perform the same role as macroinvertebrates, but in the water column, and they are relatively easy to sample quantitatively. Our major interest with zooplankton is species richness and what the proportionate abundances of various species can tell us about a lake's quality. Zooplankton are sensitive to fish predation and fish

introductions, as well as to trophic state and dominant phytoplankton, and they are the dominant animals in most lakes in terms of numbers and biomass. Because we are mostly interested in rotifers, cladocerans, and copepods, we will take an index sample by vertically towing a fine mesh Wisconsin net at the deepest point of each lake. At a subset of lakes, multiple tows will be taken and some lakes will be sampled twice to assess temporal variability.

Data interpretation will center on species richness, relative abundance of generally tolerant or intolerant taxa, and guilds. Zooplankton are typically not very speciose at any one time, with 4-15 species per tow in most small lakes (Lind 1974, Ruttner 1963).

1.5.2.4. Sedimentary diatom assemblages will be used to evaluate the extent and rate of environmental and biological change (biotic integrity) in lakes. Because of their siliceous cell walls, diatoms preserve well in most natural waters. Thus, samples collected from the top of the lake sediments integrate the diatom assemblages over the full year and the entire sediment core represents continuous deposition since a lake's formation, preserving a record of its evolution.

Diatom sample collection, preparation, and analysis methods are fairly well developed for lakes (Charles and Whitehead 1986, Dixit et al. this volume). The diatom assemblage is composed of dozens to hundreds of species, each of which has different environmental requirements. Analysis of such a speciose assemblage provides a powerful tool for interpreting status and monitoring change. A single core of bottom sediments provides sufficient material to develop diatom profiles for the entire lake. Recent developments in statistical analysis of diatom data make it possible to calibrate top sediment assemblages with current condition in a range of lakes, and then to reconstruct presettlement conditions. We may use a similarity index such as Pinkham and Pearson B (Boyle et al. 1990) to evaluate degree of historical change. The use of diatom frustule analysis in paleoreconstructions has become more widespread also. It has been used effectively to

evaluate trends in acidification, climate, eutrophication, and general pollution conditions (Agbeti and Dickman 1989, Charles et al. 1990, Lowe 1974, Stoermer et al. 1985).

In streams, algae will be scraped from stable substrates such as cobbles and snags. Although algal biomass, production, and chlorophyll-a are highly responsive to natural disturbance, algae recover quickly and are ubiquitous and easily sampled (Grimm and Fisher 1989, Steinman and McIntire 1990). As in lakes, the proportionate abundance of species will be related to known physical and chemical habitat requirements; species and family richness and relative abundances will also be assessed. Voucher slides of diatoms will be archived with a national research center, such as the Philadelphia Academy of Natural Sciences.

1.5.2.5. Trophic state indices will directly address the trophic status issues of water clarity, noxious algal blooms, and anaerobic conditions in lakes. Trophic status will be assessed primarily with Secchi disk transparency measures and concentrations of chlorophyll-a and total phosphorus. It will be necessary to measure other attributes to help interpret these basic measurements. Total nitrogen, total suspended solids, dissolved organic carbon, color, and extent of macrophyte coverage will be useful where lake conditions are not dominated by phytoplankton. These data will be converted to one or more indices such as Carlson's trophic state index (Brezonik 1984, Carlson 1977, Shapiro 1979, Walker 1984). Biological assemblage data can also be used for evaluating trophic state, although simple taxonomic aggregations of algae are ineffective (Lambou 1983).

1.5.2.6. Semiaquatic wildlife assemblages will be used to further assess biotic integrity. This indicator will focus on birds that spend a substantial portion of their lives in or near lakes. Many of these species are important recreationally (for hunting or as "watchable" wildlife), and as a group they link lakes with their watersheds and the larger region. Bird assemblages are particularly sensitive to disturbance of riparian habitats.

This indicator has been used by the U.S. Fish and Wildlife Service's Breeding Bird Survey (BBS, Robbins et al. 1986). The methodology will be modified for canoe observations in deep waters and walking in shallow waters. Data will be assessed using a modification of the response guild system of Brooks et al. (In Press). The optimal index period for birds is in the spring or early summer when the BBS occurs. This does not coincide with the mid-summer sampling period currently planned for EMAP. Because of the timing, the necessary expertise in field identification of birds, and the existing BBS protocols, these data could best be collected through an agreement with the Fish and Wildlife Service.

1.5.3. Exposure and Habitat Indicators

Historically, water body quality has been evaluated primarily with exposure indicators, such as the physical and chemical characteristics of surface waters. In EMAP, exposure and habitat indicators are intended to serve a diagnostic function when measured in conjunction with response indicators. They are used to estimate the expected natural range of ecological condition and to identify the likely causes of impairment detected by the response indicators. They may also suggest the potential for future impairment, for example, via bioaccumulation of toxics in fish that threaten the health of piscivorous animals, or loss of large woody debris which decreases stream complexity. EMAP-LS plans to use measures such as sediment toxicity, chemical contaminants in fish, biomarkers, water quality, and physical habitat as exposure and habitat indicators.

1.5.3.1. Physical habitat quality characterizes the physical conditions that may limit the biological components from reaching the full potential expected for that region and that waterbody type and size. Gorman and Karr (1978) found that habitat diversity was correlated with fish species diversity in Indiana and Panama streams and Angermeier and Karr (1984) demonstrated the importance of woody debris on fish species richness and abundance. Miller et al. (1989) listed physical habitat degradation as the primary cause for extinction of 40 North American fish taxa over the past 100 years. In some cases, physical habitat limitations are natural (floods, droughts), in others, human caused. In

either situation, physical habitat information is needed to fully interpret the response indicator data. For example, if the physical habitat score is high but the biological scores are low, then a water quality impact is suspected as the cause of the low response indicator score. On the other hand, if the physical habitat is severely degraded, then the biological component will be impaired no matter what the water quality. We hope to develop diagnostic tools that will help us to better define the relationships between physical habitat and biological scores.

Physical habitat can be grouped into five components: riparian condition, waterbody size and persistence, and habitat type and complexity (Plafkin et al. 1989; Platts et al. 1983). Riparian condition refers to the type and amount of vegetation near or over the water body and the interaction between terrestrial and aquatic ecosystems. It also includes bank stability and composition and immediate land uses. As a type of wetland, riparian areas are critical habitats for both aquatic and terrestrial organisms. Water body size includes surface area, maximum and average depth, and volume of water. For streams, volume is a function of cross-sectional area and gradient. Persistence is a relative measure of the stability of the volume of water (e.g., lake level fluctuations, frequency and regularity of stream floods, and/or droughts). Habitat type refers to a combination of water depth/speed and substrate type. Habitat types for lakes include profundal, littoral, vegetated or unvegetated shore, beach, and others. Among the stream habitat types are riffles, pools, runs, glides, and backwaters. Substrate type refers to the bottom material or to objects in the water on or among which organisms live. Habitat complexity includes the number and proportion of the different kinds of water depths/velocities, habitats, and substrates occurring in the waterbody. At a higher level of integration it includes stream channel or lake basin morphologies. Generally, these data will be collected by recording observations made along transects of the lake or stream. A research task for this indicator is to develop a set of measures that adequately characterizes each waterbody, and that produces consistent, quantitative, repeatable data.

1.5.3.2. Water quality is the suite of variables that describes the water column conditions to which aquatic biota are exposed and indicates possible chemical stresses on the biota (Hynes 1974, Warren 1971). Much of the variability in aquatic species composition and abundance not explained by physical habitat is likely to be associated with natural and anthropogenic differences in water quality. The levels of major ions, pH, temperature, and dissolved oxygen often control the chemical behavior and biotic stresses of other chemical constituents (e.g., toxins, nutrients, metals). Quantitative description of the chemical habitat is a necessary part of the diagnosis of proximal causes of impairment, and facilitates an evaluation of the linkages between human activities and biotic responses.

EMAP-LS will examine four general attributes of water quality: water type (ionic strength), acid-base status (sources of acidity), nutrient status, and redox status (possible toxic conditions). There is some overlap among the chemical species collected to assess these categories. The water type variables (e.g, Na, K, Mg, Ca, Si, SO_4, NO_3, Cl) indicate the association between water quality and soil weathering processes, and some anthropogenic disturbances. Acid-base status includes measures of hydrogen ion activity and buffering capacity. The former controls the solubility and toxicity of many chemicals, the latter describes the degree to which the waterbody resists pH changes. Anion measurements are used to assess potential sources of surface water acidity. Nutrient status addresses the supply of chemical compounds that often limit the growth of algae and macrophytes (NO_3, NH_4, total N, total P, ortho P). The redox status of waters (assessed by D.O., pH, temperature, Mn, and Fe) is a major control on the solubility, mobility, and toxicity of many chemicals including nutrients and toxic heavy metals.

For lakes, water temperature (profile), dissolved oxygen (profile), conductance, pH, and Secchi disk transparency will be measured once in the deepest part of the basin, as estimated by sonar or bathymetric maps. Epilimnetic (1.5 m) and near-bottom (for low D.O. lakes) water samples will be collected for laboratory analyses of the remaining chemicals, using methods of the National Acidic Precipitation Assessment Program (U.S.

EPA 1987a). For streams, grab samples will be taken just below the surface in approximately the deepest water.

1.5.3.3. Biomarkers taken from fish blood, gill, and liver tissues will be evaluated to diagnose early signs of stress (Cormier and Neiheisel this volume). Stress proteins, liver enzymes, and blood chemistry from five bottom dwelling fish per site will be examined. We are particularly concerned with screening for short-lived chemicals that are difficult to detect by other means. These indicators are also being examined for their ability to give early warning signals of ecosystem stress. A third reason for using biomarkers is to provide an organism-level indicator to complement the ecosystem (trophic state) and assemblage level response indicators.

We propose to evaluate biomarkers at a subset of lakes because of insufficient funds and because their sensitivity in moderately impaired and minimally impacted waters remains questionable. A second research objective is to compare biomarker sensitivity and response with those of the more typically used assemblage, ecosystem, and exposure indicators.

1.5.3.4. Sediment toxicity tests (Nebeker et al. 1984) will address possible biological exposure to toxic materials that have accumulated in sediments. EMAP is not designed to pinpoint toxic "hotspots", but should be able to characterize, on a regional basis, the relative frequency with which toxics may be impacting biotic integrity and fishability. Two liters of sediment will be removed near the deepest point of the lake and iced. We propose to conduct 10-day solid-phase lethality tests with Hyalella amphipods and 7-day solid-phase growth assays with Pimephales promelas (fish) larvae. Each test will consist of 5 replicate treatments in which 200 ml of homogenized sediment is placed in a one-liter beaker with 800 ml of clean overlying water. Controls will contain clean aquarium sand and temperatures in the tests will be maintained at 25°C. The chemical character of the control water, the physical character of the control substrate, and the use of infaunal test organisms are subjects of further investigation.

1.5.3.5. Chemical contaminants in fish (Schmitt and Brumbaugh 1990, Schmitt et al. 1990) directly assess the exposure to chemicals that fish have experienced, and express a potential health hazard for humans and the biota. Throughout this activity, we will maintain close coordination with the USFWS National Contaminant Biomonitoring program. Currently proposed analytes are: Aldrin, Arochlor, Benzo(b)fluoranthene, Benzo(k)fluoranthene, Benzo(g,h,i)perylene, Benzo(a)pyrene, Chlordane, Crysene, DDD, DDE, DDT, Dibenzo(a,h)anthracene, Dieldrin, Endrin, Fluoranthene, Fluorene, Heptachlor, Heptachlor Epoxide, Hexachlorobenzene, Hexachlorobutadiene, gamma-Hexachlorocyclohexane, alpha-Hexachlorocyclohexane, Indeno(1,2,3-cd)pyrene, Phenanthrene, Pyrene, arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, and zinc. Numerous research questions have yet to be answered for this indicator. Several of these questions revolve around the risk assessment process, such as which chemicals to target, whether to use whole fish or fillets, which species to use, which health criteria to use, which chemical analyses to use, how to assess risk of multiple chemicals, and whether to archive samples.

1.5.4. Stressor Indicators

Stressor indicators describe those characteristics, processes, or activities that are the sources of chemical, physical, or biological perturbation. Stressor indicators may include natural processes, such as drought, flooding, and fire. However, the primary intent of stressor indicators is to characterize the activities over which humans have greater control and that can be related to creating deleterious conditions.

The primary role of exposure/habitat and stressor indicators within EMAP is to identify the probable causes for impaired conditions. Thus, the assessment process for these indicators differs from that envisioned for response indicators. Except for the final assessment of ecological condition, the steps in Figure 1 may be used to assess these diagnostic indicators. After metric and index scores have been calculated, further analyses will be required to relate these indicators to the behavior of the response

indicators. Correlation of conditions in response indicators with various exposure/habitat and stressor indicators is the first level of this analysis.

Our major focus this year is on the response and exposure indicators, so our stressor indicators are much less developed. Currently, Tier 2 field sampling is planned for few of the data needed to assess the stressor indicators. Instead, these data will be collected from a variety of existing data sources, including maps, management and regulatory agency reports and databases, and the landscape characterization of the 40-km^2 hexagons made for Tier 1 (see Hunsaker et al. this volume). One issue for the surface water component of EMAP is whether the hexagon characterization is adequate, or whether the complete watersheds for the Tier 2 waterbodies need to be characterized.

1.5.4.1. Land use and land cover characterizations will be the most general stressor indicators. We plan to use land use/land cover maps, digital data bases, and remote sensing imagery as data sources. Land use will describe the most prevalent types of human-induced stress (percent and intensity or type: agriculture, urban, mining, grazing, silviculture). Omernik (1977) and Smart et al. (1985) have shown land use to be closely related to stream nutrient levels.

1.5.4.2. Human and livestock population density from maps will be examined to help us describe the general level or intensity of potential human-induced or livestock damage. Platts (1979) reviewed the impacts of livestock grazing on stream ecosystems and summarized effects on riparian vegetation, channel morphology, water quality, and fishes.

1.5.4.3. Pollutant loadings can be assessed by analyzing current pollution discharge permits for point sources, both municipal and industrial. Permit information will further describe the kinds and intensity of nutrient, chemical, and thermal stresses. This stressor also includes nonpoint loadings from atmospheric deposition and agricultural chemical application (U.S. EPA 1990b). These data are mapped at the regional and county levels.

1.5.4.4. Flow and channel modifications can be assessed from the landscape characterizations and agency records (e.g., Army Corps of Engineers), and from habitat evaluations during Tier 2 sampling. Karr et al (1983) and Simpson et al. (1982) described how these changes affect stream fishes.

1.5.4.5. Stocking, harvesting, and species introduction records from state and federal fishery agencies provide information about biological stresses in aquatic ecosystems. The intensity of management efforts will be used to evaluate the sustainability of fisheries and the possible replacement of native species. Regier and Hartman (1973) described how fisheries management practices contributed to the decline in native stocks of Lake Erie fishes and Miller et at. (1989) listed introduced species as the second most important cause of the extinction of North American fish taxa.

1.5.5. Volunteer Monitoring and Use of Existing Data

For EMAP or any monitoring program to be successful, it is necessary to have considerable public support. One way to obtain such support, and to increase the amount of data collected, is through volunteer monitoring. Volunteers may reside along EMAP stream or lake sites, or service groups and schools could adopt a set of sites for monitoring. Indicators such as Secchi depth, macroinvertebrate foraging guilds, and bird and game fish species and abundances are currently monitored by volunteers (Cummins and Wilzbach 1985, Robbins et al. 1986, USEPA 1990c). Volunteers will be subject, for the indicators selected, to the same training, methods, and quality assurance audits as paid staff.

Data can be obtained through existing state and federal monitoring programs also. Nearly all states conduct compliance and screening monitoring for water quality problems and the U.S. Geological Survey and the U.S. Fish and Wildlife Service have biological monitoring programs on scales similar to EMAP's. Although these programs lack a statistical design for national and regional extrapolation, the data should be useful if

indicators and sampling methods are comparable and quality assurance is adequate. Differences in indicators or sampling and analytical methods may prevent use of data from other agencies to assess status or trends without a considerable number of calibration studies. The EMAP grid design can also be intensified for states desiring greater statistical power for their monitoring. This was done for the California Air Resources Board in 1990 to aid in their survey of fish in Sierra Nevada lakes.

Finally, a limited number of monitoring programs are being conducted by universities, biological stations, national laboratories, and public utilities. Such monitoring tends to be site specific, but offers excellent temporal resolution and greater insight into natural processes. Spatially extensive data are available from systematic collections of museums. Data from all these sources is unlikely to be used in assessing regional status and trends, however, insights gained from such researchers will be extremely useful in diagnosing probable causes of impairment and in interpreting trends.

1.5.6. Quality Control Activities

Quality control (Table 4) is critical in a program such as EMAP-LS. The biological sampling, physical habitat assessment, taxonomic identifications, toxicity tests, landscape characterizations, and chemical analyses must be conducted by people with considerable expertise (staff from state or federal agencies, universities, or contractors). At least on a regional basis, standard sampling methods must be implemented and as methods change, the new methods must be documented and compared with the old to allow development of data conversion factors. In addition, intensive training is required to ensure that sampling and analytical protocols are understood and heeded. Because unexpected field situations will arise, methods for responding to them must be planned and a coordinator must be available for daily consultation with field crew chiefs. Field and laboratory audits and seasonal debriefings will be conducted to further assure that protocols are followed and necessary corrections are made. Quality control data will be collected, analyzed as part of the data set, and the results will be published. Quality

control data will be used to select indicators, improve sampling and analysis methods, and evaluate and improve data quality.

1.5.7. Establishing Impaired (Subnominal) Condition

The process of selecting and implementing indicators cannot be separated from the process of using the data to make statements about the condition of lakes and streams. The indicators will show that some of the waterbodies are impaired (subnominal, unhealthy). The question is one of establishing the criteria that define impaired systems, that is, for deciding that below certain index or metric scores a lake or stream is considered to be in unacceptable condition, relative to a particular endpoint of concern. Some of the approaches to establishing these criteria include (1) reference sites, (2) historical data, (3) opinions of regional experts, (4) pristine sites, (5) ecological models, and (6) the empirical distribution of indicator values.

EMAP-LS will interpret ecological condition based on a combination of these approaches but will rely heavily on a series of regional reference sites (Hughes et al. 1986, 1990, Hughes 1989, Whittier and Rankin this volume) and historic condition as described through paleolimnological reconstructions (Charles et al. 1990) and existing collections (Hocutt and Wiley 1986). This will allow us to integrate professional judgement (ours and that of regional experts), an understanding of historical or pristine conditions, and knowledge of current ecosystem research to select the least disturbed (most natural) but representative sites of a region for each water body type and size of concern. The response indicator values and the statistical variability found in the reference sites can be used to develop the expected or acceptable conditions. During pilot studies, a small set of minimally and maximally impacted reference sites will also be sampled to evaluate the responsiveness, sensitivity, variability, and redundancy of indicators.

It is expected that the waterbodies selected for Tier 2 sampling will represent the overall range of conditions for each region and that very few of the best sites will be selected.

Purposely selecting an additional set of unimpaired sites will ensure a robust model of unimpaired conditions. Landres (this volume) has also proposed a set of reference sites, judged to be as pristine as possible and incorporating ecosystem variability, to define criteria for ecosystem health and significant change. This will be very difficult in intensively farmed regions, but Marsh and Luey (1982) argued that relatively natural areas occur along streams even in the midwestern Corn Belt. It will also be useful to examine the types of stressors impinging on the Tier 2 waterbodies, and to purposely select highly impaired systems to ensure that our indicators perform as predicted.

Another option for determining whether a system is impaired is through one or more ecological models. Models may be based on field data for a key variable or set of variables, complex ecosystem studies, or laboratory toxicity tests. Empirical models that identify relationships, such as maximum fish species richness lines as a function of lake area (Matuszek and Beggs 1988, Minns 1989) or stream size (Fausch et al. 1984, Hugueny 1989), are an effective way to establish criteria. Habitat classification models and water quality criteria should be useful for assessing exposure/habitat influences. However, key exposure/habitat variables differ considerably across the nation, requiring considerable model refinement. The fact that models tend to be stressor or species specific and/or applicable to limited geographic areas will constrain their usefulness initially. However, as EMAP progresses, models should become increasingly important for refining criteria.

Selecting an impaired/unimpaired boundary based on the empirical distribution of indicator data from Tier 2 sampling sites has been suggested. For example, such critera could be chosen as the median value (or 25th percentile), or at some breakpoint in the distribution of all values. There are three problems with using a purely empirical approach such as this. First, distributions of most indicators are continuous (without clear breakpoints). Second, given the extent and frequency of diffuse pollution, there is a high probability that nearly all grid sites may be impaired. Finally, there is little ecological basis

or relevance to this approach. The impaired/unimpaired boundary must be based on some sense of the amount of change that is detrimental to an aquatic ecosystem.

Regardless of which methods we emphasize to establish the impaired/unimpaired boundary, we will use elements of all. Clearly there are natural regional differences in species distributions and environmental variables. To ignore these would be foolish, but there are also finer grained differences that govern species distributions and abundances. In lakes we have excellent historical records of environmental and biological conditions deposited in the sediments since the lakes formed. Museums contain records of historical collections and these are summarized and interpreted in state or national books (e.g. Hocutt and Wiley 1986, Lee et al. 1980). This information, together with the knowledge of expert field biologists, can be used to delineate regions, major water body types, and habitat preferences. We must use both types of historical information to estimate impairment. Historical information is essential also for developing more quantitative models of habitat-assemblage relationships. Finally, no matter how we set criteria, judgement will be involved in drawing the line between impaired and unimpaired states.

We do not expect to answer these questions quickly or easily because the definition of impaired/unimpaired conditions has social and scientific components. Our intent in EMAP-LS is to lead the discussion on how to objectively establish these criteria. For example, historical fish assemblage data provided by Ohio EPA (Figure 3) indicate significant ecoregional differences in IBI scores (X^2 = 257.38, df = 3, P <.01) and significant differences between regional reference sites and all sites in each region (X^2 = 21.23, df = 3, P <.01; X^2 = 70.94, df = 3, P <.01). If an impairment criterion were set conservatively at the 75th percentile of the reference site scores for each region, 5% of the sites would attain it. Using the 25th percentile, 36% of the Huron Erie Lake Plain (HELP) and 20% of the Western Allegheny Plateau (WAP) sites attain it. The choice of percentile must reflect our relative willingness to accept Type 1 and Type 2 errors (Green 1989, Peterman 1990).

Furthermore, evaluations among fish, macroinvertebrate, and diatom assemblages will occasionally be contradictory. Clearly, the criteria we set will impact the conclusions we reach. To reduce interagency conflict and public confusion and to maximize information exchange, we must agree on criteria and a process to evaluate them as continually more data are obtained. EMAP will produce data that will form the foundation for the assessment; we believe we also have an obligation to estimate initially where the impaired/unimpaired boundary might be. However, to have widely acceptable condition criteria, many competing social, economic, scientific, and political issues must be exami

1.6
SUMMARY AND CONCLUSIONS

EMAP-Lakes and Streams offers a method for making quantitative regional and national estimates of endpoint status and trends in surface waters. It does so through use of a population based study design and sets of response and diagnostic indicators. But these indicators will undoubtedly change. A major objective of this and future regional pilots will be to decide which indicators to discontinue or assess at fewer sites. We are cognizant of the need for change because FY-91 funding levels are insufficient to both collect data on all the indicators and monitor 50-100 lakes in the summer 1991 pilot. Projected funding will also preclude sampling all indicators when EMAP-LS is implemented nationwide at 1,600 sites annually. In addition, collection methods, measurements, and the metrics and indices calculated from them are expected to evolve as EMAP matures. Finally, we expect considerable change in how we will establish impairment, both because our methods are as yet undetermined and because biological criteria are being developed by 50 different states over the next 6 years (U.S. EPA 1990a).

1.7
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1.8
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Table 1. Candidate Indicators for Lakes and Streams

Response
Trophic State Index
Sedimentary Diatom Assemblage
Zooplankton Assemblage
Macroinvertebrate Assemblage
Fish Assemblage
Semiaquatic Wildlife Assemblage
Exposure/Habitat
Physical Habitat Quality
Water Quality
Sediment Toxicity
Chemical Contaminants in Fish
Biomarkers
Stressor
Landuse and Landcover
Human and Livestock Population Density
Chemical Use
Pollutant Loadings
Flow and Channel Modification
Introduced Species
Stocking and Harvesting Records

Table 2. Linkages between Endpoints, Measurements, Metrics and Response Indicators for EMAP-Surface Waters

Endpoints	Measurements	Metrics (provisional)	Indicators (w/provisional scoring criteria)		
No./% OF FISHABLE WATERS			FISHABILITY INDEX (0 or 1)[1]		
gamefish sp. present	sp. identification	proportionate abundance	absent (0)		
gamefish abundance	no. of individuals	catch per unit effort (CPUE)	<.002 [1/8H] (0)		
gamefish size	individual length/weight	% keepers, % trophy	<50% keepers (0)		
fish appearance	external anomalies	% anomalies	>5% anomalies		
fish edibility	toxic concentrations	consumption criteria violations	restricted >once/yr (0)		
fishery sustainable	individual weight, length, scales;	age/size structure	juveniles absent (0)		
			>50% adults stunted (0)		
	stocking records, catch restrictions	% wildfish, % keepers	<50% wildfish (0), <50% keepers (0)		
TROPHIC CONDITION					
noxious algal blooms, surface scums	pigment concentrations, visual	% blue green algae	>10% (0)[1]		
	sediment diatom sp. & abundance	% nuisance sp.	>10% (0)		
macrophytes	macrophytes	% of lake macrophyte dominated	>25% littoral zone (0)		
low transparency	Secchi depth	Secchi depth	<2m (0)		
noxious taste/odor	threshold odor	threshold odor number (TON)	TON > 10 (0)		
fish kills	hypolimnion oxygen concentrations	% depth <3 mg DO/l	>50% (0)		
trophic state	pigment concentrations; total phosphorus; Secchi depth; total nitrogen, color, turbidity	trophic state index (TSI)	TROPHIC STATE INDEX (< 30, 30-60, > 60)		
		color (Pt), turbidity (FTU)	> 165 color, >3 turbidity		
change in trophic state	sediment diatom sp. & abundance	% oligo/meso/eu/dystrophic sp.; dominant sp.; % epiphytic; % planktonic	rate of historical trophic state change (Δ TSI/YR)		
BIOLOGICAL INTEGRITY			BIOINTEGRITY INDEX		
			(1)	(3)	(5)
"dead" lakes	no. of individuals[2]	CPUE	<33%	33-67%	>67%[3]
declines in sp. richness	sp. identification[2] & no.	no. of species	"	"	"
declines in sensitive sp.	sp. identification[2] & no.	% sensitive	"	"	"
increases in tolerant sp.	sp. identification[2] & no.	% tolerant	>25	10-25	<10
increases in exotic sp.	sp. identification[2] & no.	% exotic	>10	1-9	<1
reproductive failure/population sustainability	individual length & weight (fish)	age/size structure	>1+	1+->1+(3)	0+->3+
	no. of individuals (birds)	CPUE	See CPUE above		
evidence of kills	questionnaire	kill frequency	1-5 yr	>5 yr	no kill
increased anomalies	external anomalies (fish)	% anomalies	>5	1-5	<1
decreased abundance	no. of individuals[2]	CPUE	See CPUE above		
decreased max individual size	individual length & weight (fish)	% old growth	<1	1-9	>10
historical dislocation	species & abundances (diatoms)	% similarity (Pinkham Pearson B)	<25	25-75	>75

1 0 = not fishable or undesirable trophic condition; 1 = fishable or satisfactory trophic condition.
2 Includes sedimentary diatoms, zooplankton, macroinvertebrates, fish, and birds.
3 Criteria are based largely on regional reference site values; higher scores represent higher quality sites.

Table 3. Linkages between Societal Concerns, Measurements, and Exposure/Habitat Indicators for EMAP-Surface Waters

Societal Concerns	Measurements	Indicators
Are fish safe to eat?	fish tissue toxics (metals, organics) concentrations	violation of consumption criteria >1/yr
Are waters becoming more eutrophic	P, N, Secchi, pigments, lake sedimentary diatom sp. & abundance macrophytes, DOC.	Trophic State Index (TSI) % eutrophic, mesotrophic, oligotrophic diatoms littoral macrophytes, DOC concentration
Are waters becoming dystrophic?		
Are waters becoming more saline?	Cl, TDS lake sedimentary diatom sp. & abundance	total chloride, total dissolved solids % halophilic, halophobic
Is water acidity changing?	pH, DIC, ANC, Al, SO4, NO3, C1, DOC TN, Na, K, Mg, Ca, NH4 lake sedimentary diatom sp. & abundance	ANC, anions, cations % acidophilic, acidophobic
Are waters warming?	% canopy (streams) lake sedimentary diatom sp. & abundance	% canopy (< 25, 26-50, 51-75, >75) % stenothermal, metathermal, eurythermal
Are waters becoming more turbid?	Secchi, turbidity, TSS	Secchi depth, turbidity, total suspended solids
Are sediments toxic to aquatic life?	Hyalella, Ceriodaphnia & Pimephales mortality & reproduction or growth	toxicity significantly greater than controls
What is the critical level of habitat alteration for eliminating habitat sensitive fish & wildlife?	LAKES: % littoral dominance, lake area, max. depth, level fluctuation, substrate, cover, depth variation, shoreline development, landuse, and vegetation.	Lake Habitat Quality Index (LHQI) (% regional reference sites value <25, 26-50, 51-75, >75)
To what degree is biological impairment due to natural habitat conditions?	STREAMS: Widths, depths, substrate, cover, embeddedness, flow, channel alteration/ complexity, pool/riffle/run/bend, bank stability, riparian vegetation, immediate land use.	Stream Habitat Quality Index (SHQI) (% reference sites value <25, 26-50, 51-75, >75)

Table 4. Quality Control Procedures

Fishes

Establish sampling effort criteria (duration, distance, locations, gear types and effectiveness)
Repeat sampling by different crew (10% of sites) to calculate precision and data comparability and reduce sampling error
Use standard keys and nomenclature
Preserve, label, and curate voucher specimens
Reidentify all voucher specimens to determine and improve taxonomic accuracy

Macroinvertebrates

Establish sampling effort criteria (duration, distance, locations, gear types and effectiveness)
Repeat sampling by different crew (10% of sites) to calculate precision and data comparability and reduce sampling error
Preserve, label, and curate voucher specimens
Independently re-sort, count, and identify specimens (10% of sites) to determine taxonomic accuracy and reduce identification error
Use standard local keys and nomenclature

Diatoms

Collect replicate cores (10% of sites) to estimate sampling error
Section cores at lake shore
Preserve and label section
Independently identify and count frustules (10% of slides) to determine taxonomic accuracy and reduce identification errors
Photograph all taxa and maintain a reference collection
Use standard taxonomic keys and nomenclature
Use blanks and internal standards to monitor dating of core layers

Semiaquatic Wildlife (birds)

Establish sampling effort criteria (duration, distance, locations, time of day)
Repeat sampling by different person (10% of sites) to estimate precision, data comparability, and taxonomic accuracy and reduce sampling error

Toxicity Tests

Establish minimum number of organisms per trial
Establish minimum number of replicates per test
Establish restart criteria for control mortality or container effect
Establish minimum number of young required for valid reproductive test
Establish minimum level of growth required for valid growth test
Establish minimum number of deaths required for valid mortality test
Use control charts to monitor replicate precision

Table 4 (continued)

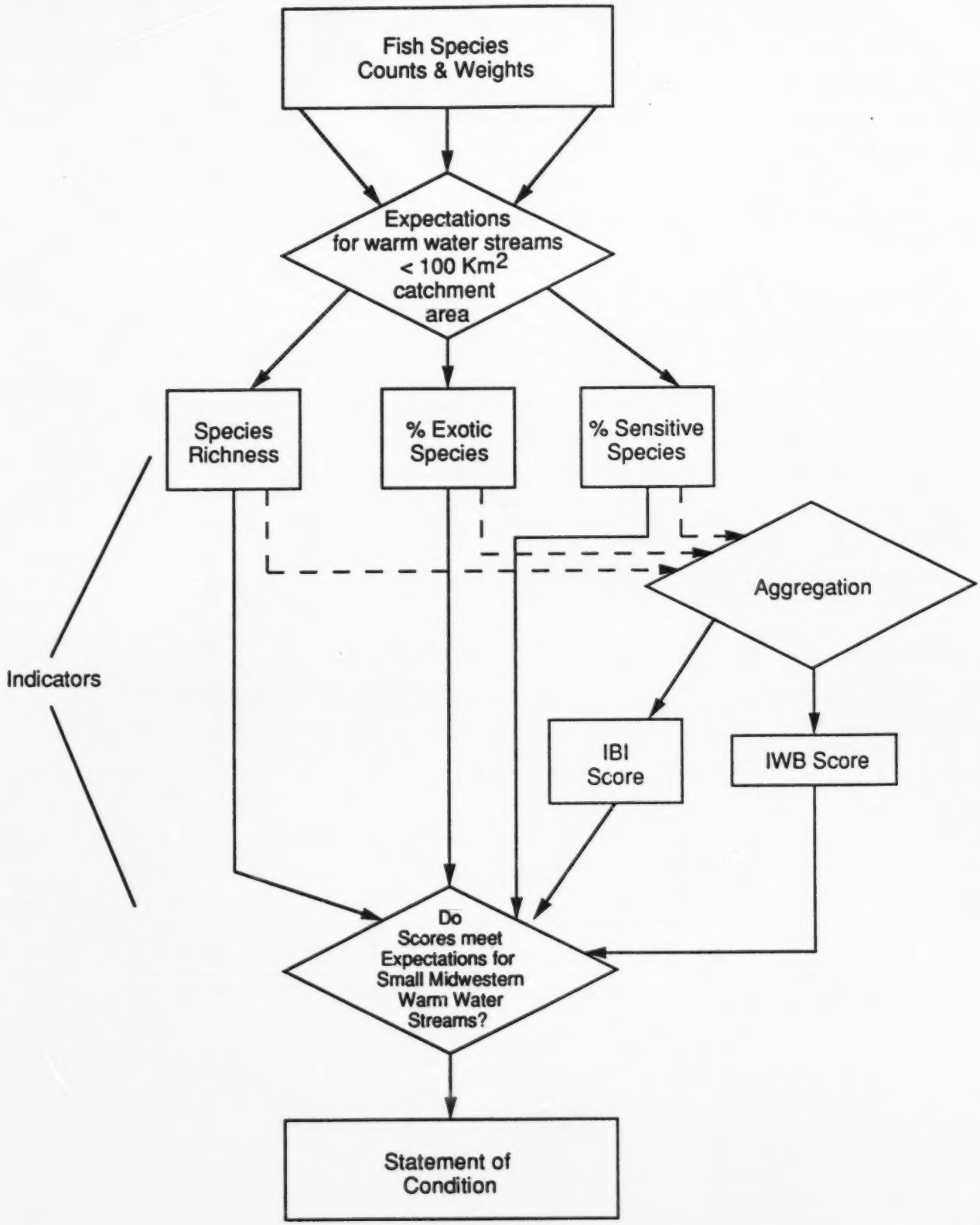
Water Quality
Calibrate instruments with independent standards
Use control charts to monitor instrument performance
Standardize collection, handling, and processing procedures
Preserve and label samples
Independent review of records on site
Use field blanks and audit materials on 1% of samples to estimate laboratory accuracy
Repeat sampling of 10% of sites by different crew to estimate precision and data comparability
Standardize laboratory preparation, calibration, and analysis
Use blanks, standards, spikes, and control charts on 1% of samples to improve accuracy and precision
Review data immediately after analysis
Reanalyze poor quality data and outliers
Physical Habitat
Calibrate instruments
Use control charts to monitor instrument performance
Standardize assessment procedures
Establish precision levels for each variable
Use a standard field sheet
Establish sampling effort criteria (distance, location, number of transects)
Independent review of records on site
Repeat sampling of 10% of sites by different crew to determine precision and data comparability and reduce sampling error

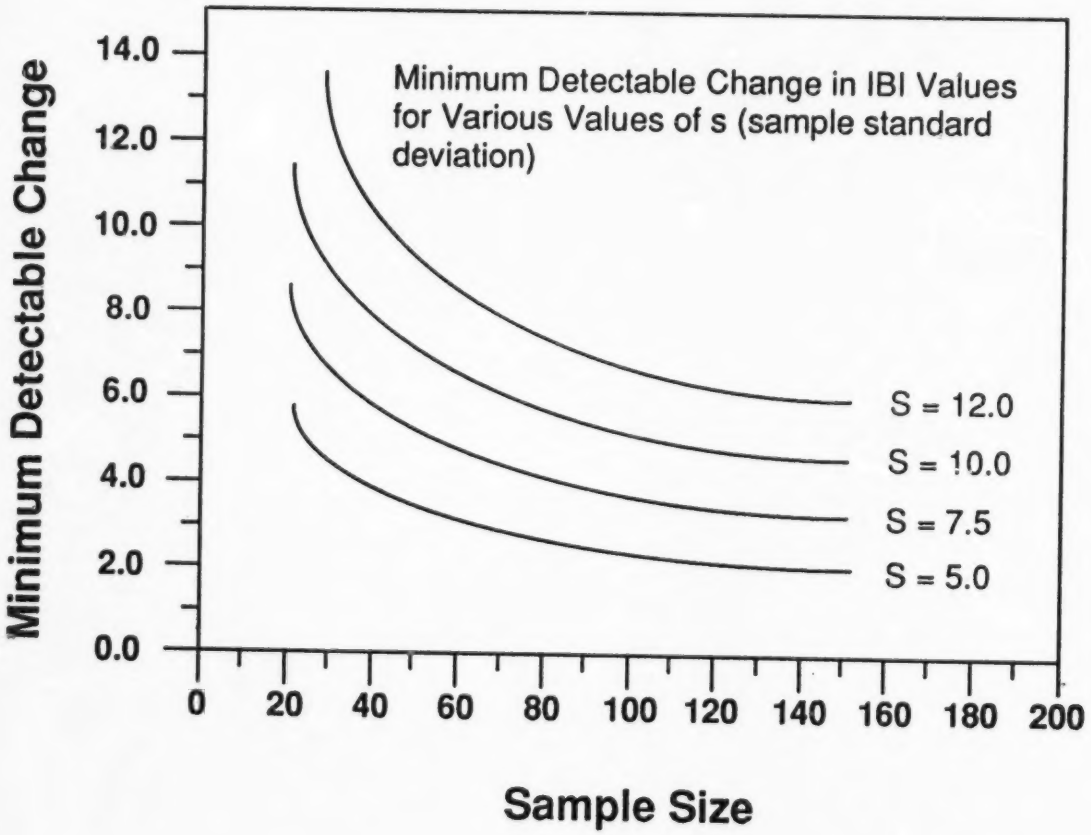
FIGURE CAPTIONS

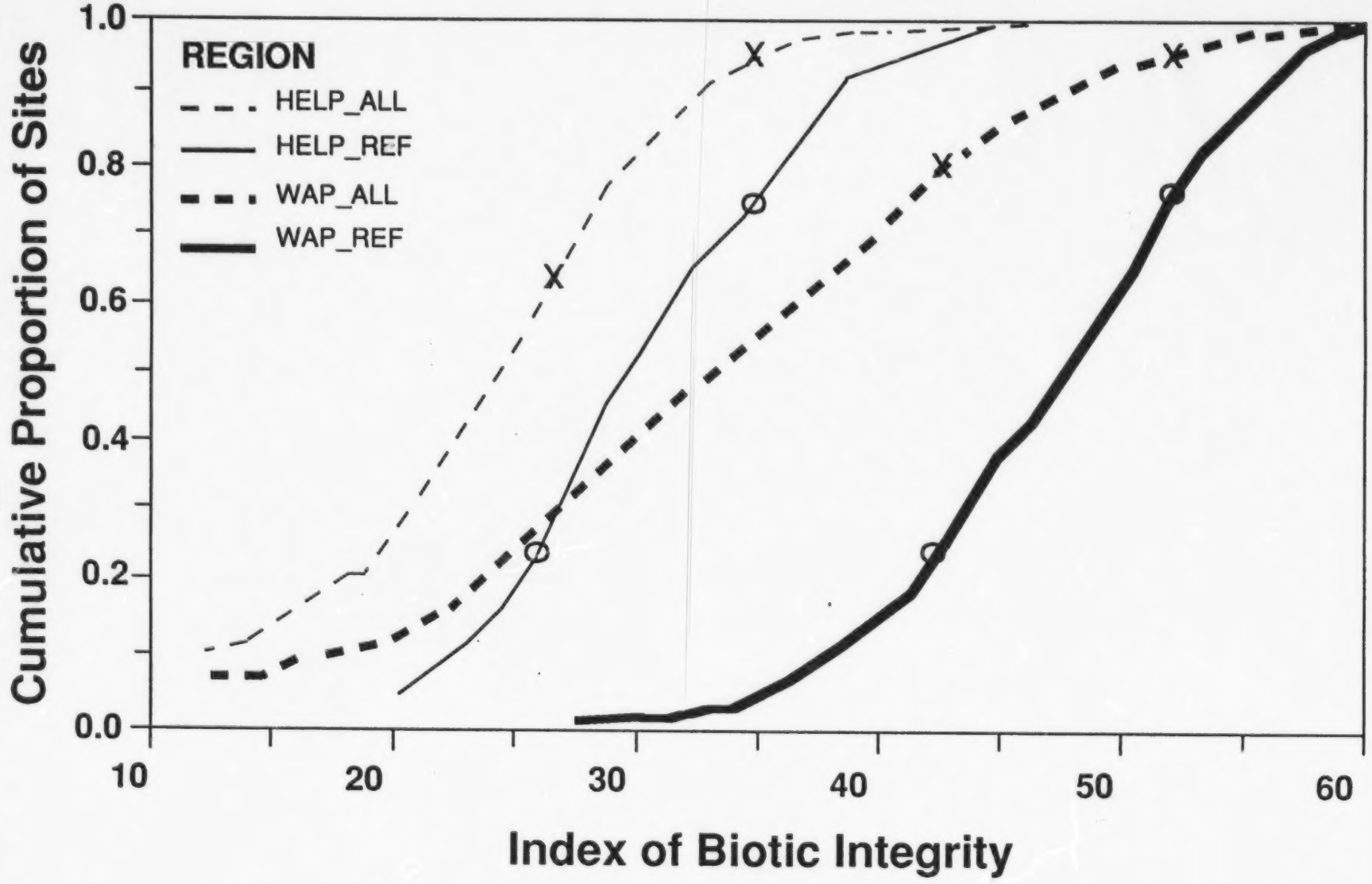
Figure 1. The conceptual process for proceeding from measurements to indicators to assessment of condition, with examples for fish assemblages. IBI = index of biotic integrity; IWB = index of well being. Note that rectangles represent numerical statements, while diamonds represent evaluations or judgements; both metric and index scores may be indicators.

Figure 2. Nomogram for determining minimum detectable change in Index of Biotic Integrity scores based on Ohio EPA data (after Loftis et al. 1989). Based on these results, we should be able to detect a change of 4-6 IBI units with a sample size of 50 and a standard deviation of 5.0-7.5.

Figure 3. Delineation of the impaired/unimpaired boundary for the index of biotic integrity (IBI) based on the 25th and 75th percentiles of Ohio EPA regional reference sites. The y-axis represents the cumulative proportion of sites with IBI scores below a given value. O = regional reference site values for IBI at the 25th and 75th percentiles; X = percentiles of sites above or below those reference values. In the Huron Erie Lake Plain (HELP) region of northwest Ohio, 64% of all sites are impaired if the boundary is set at the 25th percentile; 95% are impaired at the 75th percentile. In the Western Allegheny Plateau (WAP) region of southeast Ohio, 80% of all sites are impaired at a 25th percentile boundary and 95% at the 75th percentile.


Fish Species
Counts & Weights
Expectations
for warm water streams
< 100 Km2
catchment
area
Species
Richness
% Exotic
Species
% Sensitive
Species
Aggregation
Indicators
IBI
Score
IWB Score
Do
Scores meet
Expectations for
Small Midwestern
Warm Water
Streams?
Statement of
Condition


Minimum Detectable Change in IBI Values for Various Values of s (sample standard deviation)
S = 12.0
S = 10.0
S = 7.5
S = 5.0
Minimum Detectable Change
14.0
12.0
10.0
8.0
6.0
4.0
2.0
0.0
0
20
40
60
80
100
120
140
160
180
200
Sample Size


REGION
HELP_ALL
HELP_REF
WAP_ALL
WAP_REF
Cumulative Proportion of Sites
1.0
0.8
0.6
0.4
0.2
0.0
10
20
30
40
50
60
Index of Biotic Integrity